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Ecological sustainability in policy assessments: A wide-angle view and a close watch☆

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ABSTRACT

We give a perspective from two practitioners on some of the challenges of addressing sustainability concerns in environmental policy assessments. We focus on the ecological dimension of sustainability, which is closely related to the concept of "ecosystem resilience." First, we discuss several recent benefit–cost analyses conducted by EPA that illustrate many of the practical difficulties analysts have faced when attempting to assess the ecological benefits of proposed regulations. Next, we discuss the importance of increased coordination of policy assessments among offices and agencies that traditionally operate more or less independently. We conclude by using a stylized model to illustrate how using an "adaptive management" approach to designing and evaluating policies can help to avoid some of the limitations of standard policy assessments highlighted in this special section of *Ecological Economics* and elsewhere.

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> *It is most helpful to analyze carefully the consequences which are likely to result from the alternatives between which we have to choose. For only if we can visualize these consequences in a concrete and practical way do we really know what our decision is about; otherwise we decide blindly.*— Karl Popper (1945)

1. Introduction

Much of the research by economists on the topic of sustainability focuses on the relationship between economic efficiency, the fundamental criterion for benefit–cost analysis (BCA), and intergenerational fairness. For example, Pezzey (1997), Heal (1998), Stavins et al. (2005), among others, have clarified the conceptual differences between intertemporal optimality, efficiency, and sustainability, and shown that dynamically efficient growth paths need not automatically satisfy a sustainability (non-declining utility) constraint. Based on this observation, some suggest modifying BCA so it can incorporate sustainability concerns directly. For example, Howarth (2007-this volume) seems comfortable with seeking economic efficiency, so long as the policies incorporate a mechanism to guarantee that future generations will receive their "fair share of the ensuing net benefits" from any irreversible reductions of natural capital stocks. Others suggest supplementing or replacing BCA with other modes of decision analysis. For example, Norton (2005, 2007-this volume) recommends replacing BCA with "adaptive management," a form of iterative and deliberative decision making that would more fully involve stakeholders in a process of learning by doing.

In addition to the intergenerational fairness dimension of sustainability, there is growing interest among economists in

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the ecological dimension of sustainability – the long-term viability of biological resources and ecosystem processes and their contribution to ecosystem services – and how this is treated by BCA. In this paper, we focus mainly on this ecological dimension of sustainability, which is closely related to the concept of "ecosystem resilience," a natural system's ability to withstand stress before collapsing to a less desirable state (Holling, 1978). As Perrings (1998) notes, "the economic value of a system in some state depends on its ability to maintain the flow of goods and services for which it is valued given the shocks or disturbances it faces."[1]

The U.S. Environmental Protection Agency (EPA) has recognized a need to improve its ability to assess the potential ecological benefits of its policies (EPA, 2006a), and effects on ecosystem resilience can be thought of as one of the often neglected categories of impacts in this area.[2] Accordingly, conducting assessments that better reflect ecological complexity should lead to better policy outcomes by increasing resilience, thereby helping to avoid surprises and irreversible losses.

The remainder of the paper is organized into five sections. In Section 2, we briefly discuss the ecological dimension of sustainability and in Section 3 we draw upon our experience as practitioners to discuss the role that ecological benefits have (or have not) played in some recent policy assessments. In Section 4, we discuss the need for a "wide-angle view" of environmental problems and potential policies to address them, through increased coordination among agencies and offices that typically operate independently, to help overcome limitations associated with the conventional piecemeal approach to policy assessment. In Section 5, we point out a second way that ecological sustainability might be addressed in policy assessments by expanding on Norton's (2005, 2007-this volume) recommendation for a more adaptive approach to designing and implementing environmental regulations. This involves keeping a "close watch" on environmental conditions and policy outcomes and adjusting regulatory or management controls as conditions change. We work through a simple numerical model that illustrates how an adaptive policy approach, where policy controls are continually adjusted to match current environmental conditions, can lead to greater economic efficiency and ecological sustainability than a static approach. These points highlight just some of the challenges for researchers and policy analysts who would respond to increasing calls to assess the effects of proposed policies on sustainability. Section 6 concludes by discussing some practical aspects of our recommendations for coordinated policy assessments and adaptive management.

[1] The Resilience Alliance (www.resalliance.org) is an interdisciplinary organization that has embraced this perspective and seeks to address it with an active research agenda (Perrings, 2006).

[2] Because of this neglect, how these benefits stack up against human health benefits, the traditional focus of assessments at EPA, is very much an open question. In our experience, conjectures on this score can vary widely among analysts. To some it is obvious that ecological benefits would get lost in the typical measurement error of human health benefits. To others it is just as obvious that ecological benefits may easily be of the same order of magnitude as human health benefits or more.

2. Ecological sustainability

Ecology often emphasizes the complexity and variability of natural systems. Not only do natural systems operate at multiple scales, but their emergent properties can make predicting how an insult at one level ultimately will affect the system as a whole a daunting proposition. The proximate effect of a stressor at its source (e.g., a toxic loading to a waterbody that kills some fish) may ramify into unexpected effects elsewhere and at higher scales, where additional stressors may come into play. Effects can be transmitted from one ecosystem to others, both by the exchange of (abiotic) material and energy between them and by species migrations. Thus, although surprise is something to avoid when implementing policies, particularly when the result can be an irreversible loss of a valued ecological endpoint, it may arise more often than necessary if the natural complexities of ecological processes and ecological-economic interactions are not properly accounted for in policy assessments.

For example, a common simplification in policy assessments when attempting to value ecological effects is to assume that overall impacts are proportional to the anticipated changes in the emission rates or ambient concentrations of the pollutants or possibly limited to acute mortality rates at the level of the organism. In reality, changes at the population, community, or ecosystem levels of organization may differ substantially from the more readily quantified acute mortality of individuals due to non-linear interactions within and among species and other ecosystem processes. While the complexity of these processes makes their characterization in policy assessments difficult, ecologists have found that informative ecological forecasts (Clark et al., 2001) and models of ecological-economic interactions that are relatively simplified but still include the key non-linear relationships (Perrings, 1998) sometimes are possible.

Because of their varying perspectives, methodological approaches, and other disciplinary biases that often separate ecologists from economists, ecologists may not have felt entirely comfortable or welcome to participate fully in policy assessments, which are often led by economists. In light of their tendency to "constantly unearth complexity," ecologists are naturally uncomfortable with the historic lack of effort towards accounting for uncertainty in BCA, which leads to a false sense of precision in the results (Dovers et al., 1996). Others involved in the assessment process may consider ecologists' contributions too tentative to influence recommendations. The situation may be changing, however, with calls for ecologists to play a more integral role in policy assessments being heard more often. For example, on the need to accurately characterize the key ecological processes before conducting a valuation exercise, one prominent agricultural economist recently has suggested that ecologists should take greater initiative in the policy assessment process by "locking economists out of the room" until ecological processes are sufficiently understood (Doering, 2007).[3]

[3] Locking people out of the room probably is not the best way to motivate more meaningful collaborations between ecologists and economists, but one can appreciate the spirit of the recommendation without taking it too literally.

3. Policy assessment

Environmental regulatory agencies have not been unaware of growing concerns about sustainability, and in many cases have supported important related research intended to directly inform policy assessment practices.[4] By "policy assessment," we mean the two-stage process of identifying the policies that can address the environmental problem at hand and conducting an evaluation exercise to help decide which among the options is best. The latter stage typically consists of (or includes) a BCA. The practical point of connection between sustainability – especially the ecological dimension of sustainability that is the focus of this paper – and policy assessment is whether and how ecological dynamics, uncertainty, and irreversibilities are accounted for in the evaluation.

Limiting our inquiry to water regulations in light of their more direct connection to ecological benefits, we examined a convenience sample of seven recent policy assessments conducted in support of EPA regulations. These seven were chosen because they all are relatively recent, the regulations were economically significant, and all the assessment reports are publicly available.[5] Table 1 lists the seven regulations and briefly summarizes their treatment of ecological benefits.

In the first three assessments, the anticipated change in pollutant loadings was used to estimate a change in ambient concentrations in the water column. Where this anticipated change would cause streams that currently exceed aquatic life water quality criteria (AWQCs) to no longer exceed those thresholds, the difference in the willingness-to-pay (WTP) by anglers for fishing in un-contaminated and contaminated streams was assigned to the local angler population. While these estimates were distinguished from human health benefits and sometimes referred to as ecological benefits in the assessment, they had no bearing on how fish populations or catch rates would be affected; they were based on anglers' preferences for catching uncontaminated versus contaminated fish. So it may be more accurate to characterize these as another instance of human health benefits rather than as an ecological benefit. Non-use values were also estimated in the sense that they assumed to be fifty percent of the use values. This assumption was based on Fisher and Raucher's (1984) review of valuation studies conducted between 1974 and 1983. However, we would expect the relationship between use and nonuse values for ecological endpoints to be highly context dependent, so the assumption of a constant proportional relationship generally will not be appropriate. This seems to be an especially severe simplification in this case since the estimate of use value was not directly related to stock sizes, but rather to body burdens of contaminants that may have posed a human health risk if consumed in excess but may or may not have posed a significant risk to the fish themselves.

The next three assessments relied on an index – the RFF water quality ladder[6] – to link the estimated changes in concentrations with estimates of WTP for water quality

Table 1 – Recent policy assessments conducted by U.S. EPA

Regulation and target stressor	Translation of direct impacts to ecological benefits
Pulp and paper (EPA, 1997): Effluent discharges from the pulp, paper, and paperboard industry	Changes in in-stream concentrations were quantified using a hydrologic simulation model. Recreation benefits were estimated by assigning anglers' estimated incremental WTP for a contaminant-free fishery (Lyke, 1993) to streams no longer exceeding AWQCs.
Pharmaceuticals (EPA, 1998): effluent discharges from the pharmaceuticals industry	Non-use values estimated as 50% of recreation values. Same as pulp and paper.
Iron and steel (EPA, 2002a): effluent discharges from the iron and steel industry	Same as pulp and paper.
Concentrated animal feeding operations (EPA, 2002b): effluent discharges from large animal feeding operations	Recreational and non-use benefits: estimated changes in in-stream concentrations were mapped to a multi-metric index of water quality and assigned a fraction of an estimated total economic value (Mitchell and Carson, 1984).
Construction and development (EPA, 2004a): effluent discharges from the pulp, paper, and paperboard industry*	Same as concentrated animal feeding operations.
Aquaculture (EPA, 2004c): effluent discharges from the aquaculture industry	Same as concentrated animal feeding operations.
Cooling water intake structures (EPA, 2004b): impingement and entrainment of aquatic species by cooling water intake structures of existing large power plants	Monitoring data on fish mortality at various life stages converted to "adult equivalents." A constant fraction of mortality reductions assumed to be caught by anglers, increasing their catch rates. WTP estimated using regional random utility models. Non-use values not estimated.

* This regulation was rescinded by the Administrator and never went into effect.

[4] For example, in 1994 EPA funded an Association of Environmental and Resource Economists workshop to consider this very topic (Hartwick, 1995). The most recent example is the EPA workshop in 2006 that inspired this special section of *Ecological Economics* (http://www.epa.gov/sustainability/econforum/index.htm).

[5] A regulation is "economically significant," and therefore requires a formal benefit–cost analysis, if it is expected to have an economic impact of $100 million per year or more (President, 1993).

[6] The ladder was a synthesis of research conducted between 1955 and 1978 (Vaughan, 1986).

improvements. This was considered a significant advancement since it attempts to relate changes in physical and chemical water quality conditions to an integrated measure of aquatic ecosystem health. However, this approach also has its limitations. First, the ladder combined and mapped a limited number of water quality measures onto a continuum of recreational uses. Second, this single dimension of recreational uses was assumed to represent what is likely a multidimensional change in ecological conditions. Third, the associated WTP estimates were based on a large-scale (national level) survey on the value of improving water quality in all freshwater bodies in the U.S. (Mitchell and Carson, 1984), which may be much less accurate when applied to incremental water quality changes at a local scale.

The final regulation we consider was designed to reduce the adverse environmental impacts of cooling water withdrawals by large power plants. This regulation was qualitatively different from the others on our list in that it did not deal with a pollutant being introduced into the environment, but rather the withdrawal and destruction of organisms from it. The anticipated direct effect of the regulation was to reduce fish mortality levels, mainly at the egg and juvenile life stages. An extensive qualitative discussion of the potential indirect effects of this mortality at other trophic levels was included in the assessment, particularly in terms of the impact of forage fish losses on commercial and recreational fisheries and fish losses on piscivorous birds. Although the use of "trophic transfer coefficients" was discussed, a quantitative ecological model ultimately was not used as a means to scale up from estimates of direct losses. In its quantitative evaluation of ecological impacts and economic values, EPA estimated short-run changes in commercial and recreational fishery harvests from anticipated direct mortality reductions, rather than the long-run changes in average harvest levels and stock sizes.

This brief review suggests that only a limited accounting of ecological benefits was performed in each of the seven assessments. Ecological effects were assumed to correspond to the relative changes in the magnitude of the stressors being regulated or limited to short-run, direct impacts on select species. To monetize the benefits, a simplified benefit-transfer approach typically was used: A willingness-to-pay estimate for the most similar ecological endpoint available in the valuation literature was multiplied by the anticipated proportional change in the magnitude of the stressor and summed over all affected households.

While a certain amount of simplification always will be necessary in any assessment, further effort is nevertheless needed to incorporate ecological research into policy assessments, both directly to reduce oversimplification and indirectly to substantiate the necessary simplifications that will remain. In our (admittedly idiosyncratic) experience and even in the selective review above, we see efforts to describe and quantify the potential ecological benefits of EPA's regulations increasing over time, and as of this writing attention to these issues seems to be at a high point (EPA, 2006a). We now turn to two ways to capitalize on this increased attention to ecological sustainability by improving policy assessments. The first relates to conducting BCA in a forward-looking manner; the second to considering policy options that are flexible.

4. Institutional compartmentalization

In this section we focus on a ubiquitous feature of the institutional setting within which policy assessments are conducted that can greatly compromise their overall effectiveness: institutional compartmentalization. For example, EPA's enabling legislation consists of multiple statutes that created distinct and semi-autonomous offices, each of which are responsible for developing and implementing regulations in its particular domain. As a result of such compartmentalization, both within and across agencies, environmental policy assessments often consider a single source of environmental stress at a time. Thus, a particular set of policies may be developed to address only a fraction of the harm to a valued ecological endpoint. However, without a sufficiently expansive view of the current overall status of each affected endpoint, the piecemeal approach to policy assessment that naturally follows from compartmentalization will likely overshoot or undershoot the optimal level of regulation. The potential for over-regulation is widely appreciated by economists. Due to unavoidable resource constraints there must eventually come a point where independently proposed policies will effectively serve as substitutes for each other (or crowd each other out). However, if analyzed separately each proposal may appear to be welfare enhancing on its own because the interactions with other proposed policies are ignored. Thus, each proposal may be adopted on its own (individually assessed) merits when in fact only a subset of the policies if adopted together are welfare improving (Hoehn and Randall, 1989).[7]

However, compartmentalization also can lead to another error in the opposite direction, in part due to production functions for ecosystem services that are convex–concave, rather than strictly concave throughout, which can result from thresholds or synergistic interactions between multiple stressors (Dasgupta and Maler, 2003). Consider in Fig. 1 some ecological endpoint, such as a species population, that has collapsed as a result of a single stressor generated by multiple sources. The status quo is at the origin, where the sources are unregulated. The ecological benefits generated by abatement of the stressor is representated by the sigmoidal curve B_2. While small movements away from the origin are welfare reducing (perhaps because of a threshold in the stressor-response function), a sizeable shift to an amount of overall stringency beyond the point where the B_2 and C curves cross (where total cost equals total benefits) is welfare enhancing.

The benefits of any particular policy for reducing harm to an ecological endpoint are contingent on what else has been — and what remains to be — accomplished in terms of addressing environmental problems. However, the baseline policy mix that defines the states of the world that are compared in policy assessments typically is backward looking only, accounting for policies already in place but not for other policies that may be implemented by other "compartments" (agencies or offices). If in any one compartment no single policy that can reach the point where net benefits are positive

[7] This is closely related the concern about stated preference estimates that led to the use of the "scope test."

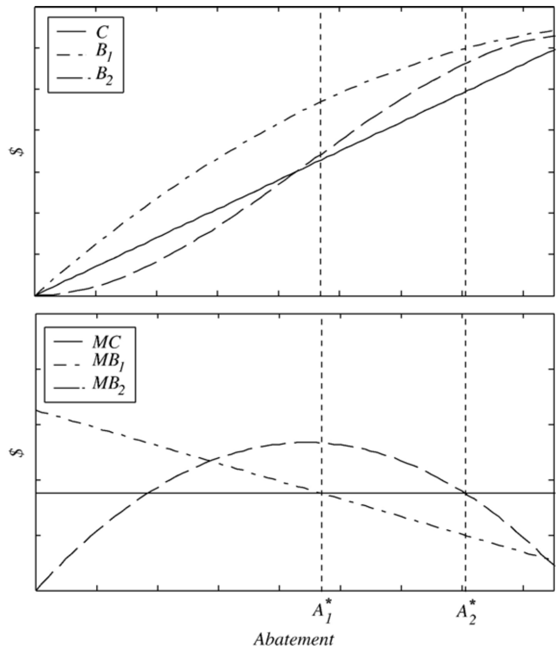

Fig. 1 – The top panel shows total cost and benefits curves; the bottom panel shows the associated marginal cost and benefits curves. With concave benefits (B_1), any incremental increase in abatement starting from the origin produces net benefits. With concave–convex benefits (B_2), a large increase in abatement from the origin (to the point where B_2 crosses C) is required to produce net benefits. Net benefits are maximized at the points A_1^* and A_2^* for the convex and concave–convex benefit functions, respectively.

is designated for consideration, then the non-convexity of the benefits curve may mean that no single policy will pass a benefit–cost test on its own.

One possible way to address this situation is to take a "wide-angle view" of the affected ecological endpoints by considering the joint effect of policies yet to be implemented that address various aspects of the harm to the endpoint. Coordination with other offices and agencies on such an assessment may reveal that one or more policy "sets" (combinations of one or more policies) will pass a benefit–cost test if all members of the set are adopted. In other words, sometimes multiple policies can act as complements, increasing returns over some range, rather than as substitutes. This is the case in Fig. 1, where a move from the origin to A_2^* is welfare maximizing given the concave–convex benefits function B_2. For example, suppose one agency considers reducing point source pollution and another agency considers reducing non-point source pollution to the waterbody where our valued species population in Fig. 1 has collapsed. Suppose further that costs are additive, but the pollution concentration threshold for species recovery is below the level that either of these policies would achieve on its own. If the agencies do not coordinate their assessments, this possibility may not be uncovered. Thus, in the face of such threshold effects the result of the compartmentalized, piecemeal approach to policy assessment can lead to under-regulation rather than over-regulation, a bias toward the status quo.

In principle, eliminating this bias when trying to identify which policy to choose to address a particular source or stressor requires first identifying the mix of feasible policies that generates the highest net benefit (the greatest flow of valued ecosystem services) from the affected ecological endpoints, considering all the stressors and sources affecting these ecological endpoints. The policy within that optimal mix that would address the stressor or source to which the original policy question referred is the one to select. Somewhat more formally, let S be the set of stressors adversely impacting an ecological endpoint, each managed by a different compartment, and let $R \subseteq S$ be the subset that includes those stressors managed by a single compartment to which a policy question currently on the agenda relates. Of all the policies Z yet to be implemented, there exists a subset $Z_S \subseteq Z$ that maximizes welfare. If $Z_R \subseteq Z$ is the subset of policies that address R, then the policy to select for R is $Z^* = Z_S \cap Z_R$. Such an assessment requires considerable interaction among compartments, and achieving this socially optimal outcome requires all compartments, when their regulatory turn arises, to adopt the policy in Z^* that pertains to them.[8] Assuming they are seeking the socially optimal outcome, the compartment will do so. As new information comes to light about the stressors in S, it may be optimal to adjust policies already implemented. This suggests the need for a flexible approach, which is the topic of the next section.

5. Adaptive management

Considering the complexity and natural variability characteristic of ecosystems, environmental policies may be enhanced by developing evaluation methods and institutional arrangements for making regulations more flexible, or "adaptive." A variety of definitions of adaptive management can be found in the literature (e.g., Holling, 1978; Walters and Hilborn, 1978; Walters, 1986; Lee, 1993; Norton, 2005). These various definitions are broadly similar, but different authors emphasize different aspects of the approach. The main themes that appear in most definitions of adaptive management include: (1) continual communication between stakeholders, decision-makers, and analysts throughout the assessment (and possibly implementation) process, (2) explicitly accounting for uncertainty, (3) active learning from management outcomes and surprises, and (4) frequent adjustment of management controls in response to new information.

In this section we develop a model of an EPA-type regulatory decision situation to illustrate how an adaptive management approach to policy assessment can avoid some of the limitations of standard policy assessment practices highlighted by the other authors in this special section and lead to greater ecological sustainability. The model is intentionally highly

[8] This is not as restrictive as it might first appear: as succeeding policy questions arise, the optimal policies to address them turn out to have already been correctly identified in ZS. Furthermore, note that the increased level of coordination that is suggested here also would help resolve the problem of too many proposals passing the benefit–cost test highlighted by Hoehn and Randall (1989).

Table 2 – Outcomes of unregulated, static, and adaptive policies, using baseline parameter values

	Unregulated	Static	Adaptive
(Average) pollution load, q	13.33	11.81	13.20
Expected present value of market benefits, M	2255.7	2226.1	2245.6
Probability of survival for 200 periods, P_{200}	0.4852	0.6445	0.9890
Expected present value of non-market benefits, N	1507.1	1574.7	1683.2
Expected present value of total benefits, V	3762.8	3800.7	3928.8

Calculated using: $a=10$, $b=0.5$, $c=0.25$, $\rho=0.03$, $K=100$, $\upsilon=50$, $\sigma_x=0.75$, and $\sigma_y=1.5$.

stylized so we can focus on the key concepts and the basic calculations. Consider an ecological endpoint (e.g., a valued species population) that reproduces rapidly but is vulnerable to pollution from an ongoing economic production activity and can be extirpated if the impacts of pollution are too severe.[9] To keep the dynamics simple, we assume that the stock size at the beginning of a time period always equals the carrying capacity K unless the impact of pollution in the previous period drove the stock to zero:

$$N_{t+1} = \begin{cases} K, & K - x_t q_t > 0 \\ 0, & K - x_t q_t \leq 0 \end{cases} \tag{1}$$

where q_t (the control variable) is pollution loadings, and x_t (a stochastic exogenous variable) is the impact of pollution (the number of individuals killed per unit of loadings) in period t. Suppose x_t is serially uncorrelated and has probability distribution $f(x)$ (which we assume to be lognormal in this example). The probability that the stock survives through period t is $s_t=\Pr[x_t<K/q_t]=\int_0^{K/q_t} f(x)dx$.

Next, assume that pollution loadings are directly proportional to economic output (so we can treat loadings and output interchangeably), and let inverse aggregate demand be $a-bq$ and let the marginal costs of production be cq (i.e., textbook linear supply and demand curves). This gives total consumer and producer surplus as a function of loadings: $\pi(q)=aq-q^2(b+c)/2$. We also assume there is an existence value for the survival of the stock, υ. The expected total (market plus non-market) net present value in period t of a production policy (i.e., a series of loadings levels q_t, q_{t+1}, q_{t+2},…) is

$$V_t = \pi_t + \upsilon s_t + e^{-\rho}(\pi_{t+1} + \upsilon s_t s_{t+1}) + e^{-2\rho}(\pi_{t+2} + \upsilon s_t s_{t+1} s_{t+2}) + \cdots, \tag{2}$$

where ρ is the discount rate.

We will consider three policy options: no regulation, static regulation, and adaptive regulation. If the economic activity is unregulated, production will occur at the market equilibrium level, i.e., where the flow of market benefits $\pi(q)$ is maximized. If a static regulatory approach is used, the regulator will set a cap on the loadings, q_S, not to be exceeded in any time period, that maximizes the expected value of Eq. (2). (This static approach is our version of what Bromley describes as standard practice and criticizes as leading to "policy lock-in.") If an adaptive approach is used – where the loadings cap is adjusted each period according to current environmental conditions – then a policy function must be designed to determine the optimal loadings cap in each period conditional on the measured (or predicted) pollution impacts. If environmental conditions are worse than average in a period, the loadings cap will be tightened, and vice versa.

Now suppose that the regulator can measure the environmental conditions and generate a prediction (albeit an imperfect one) of the pollution impact at the beginning of each period before the loadings cap in that period must be set. Specifically, suppose the measurement instrument gives $y_t=x_t+\varepsilon_t$, where the measurement error ε_t is distributed normally with mean zero and variance σ_y^2.[10] Since the adaptive regulator will adjust the loadings cap each period based on the predicted pollution impact, the expected probability of stock survival will depend on both the accuracy of the measurement instrument and the natural environmental variability; specifically, $s_t = \int_0^{K/q_t} h(x|y)dx$ where $h(x|y) = l(y|x)f(x)/\int_x g(y)$ is the posterior distribution for x (calculated using Bayes rule) after observing y, where $l(y|x)$ is the likelihood of observing y if the true value is x and $g(y)=\int_x l(y|x)f(x)dx$ is the unconditional probability density function for y.

To determine the optimal adaptive policy function, we must first manipulate Eq. (2) into a form that can be solved using dynamic programming methods (Bellman, 1957). To do this we can break it into two parts – market and non-market value components – and write each part in recursive form. Thus, $V_t=M_t+N_t$, where $M_t=\pi_t+e^{-\rho}M_{t+1}$ and $N_t=s_t(\upsilon+e^{-\rho}N_{t+1})$.

The expected present market value of the policy in any period is the total surplus in the period plus the expected present market value in the next period multiplied by the discount factor,

$$M_A(y) = \pi(y, q_A(y)) + e^{-\rho} \int_y M_A(y)g(y)dy, \tag{3}$$

and the expected present non-market value of the policy in any period is the existence value of stock survival in that period plus the expected present non-market value in the next period (conditional on stock survival) multiplied by the survival probability and the discount factor,

$$N_A(y) = s(y, q_A(y))\left(\upsilon + e^{-\rho}\int_y N_A(y)g(y)dy\right). \tag{4}$$

[9] For example, think of a locally rare aquatic species that may be exposed to algal blooms resulting from non-point source agricultural pollution. Or think of cooling water withdrawals from a river whose impacts on a local fish population depend on the river flow, which varies naturally from year to year.

[10] We might think of y as some leading indicator of x; e.g., phytoplankton community composition may be a leading indicator of the potential for toxic algal blooms (Paerl et al., 2003), or variability in lake-water phosphorus might be a leading indicator of a potential shift to eutrophic status (Brock and Carpenter, 2006). In a more realistic model the stock size also would be stochastic and subject to measurement uncertainty. We have assumed away these complications with our simplified equation for stock dynamics in Eq. (1) – which might be thought of loosely as approximating a logistic growth model with high r – but the general principles illustrated here still will apply.

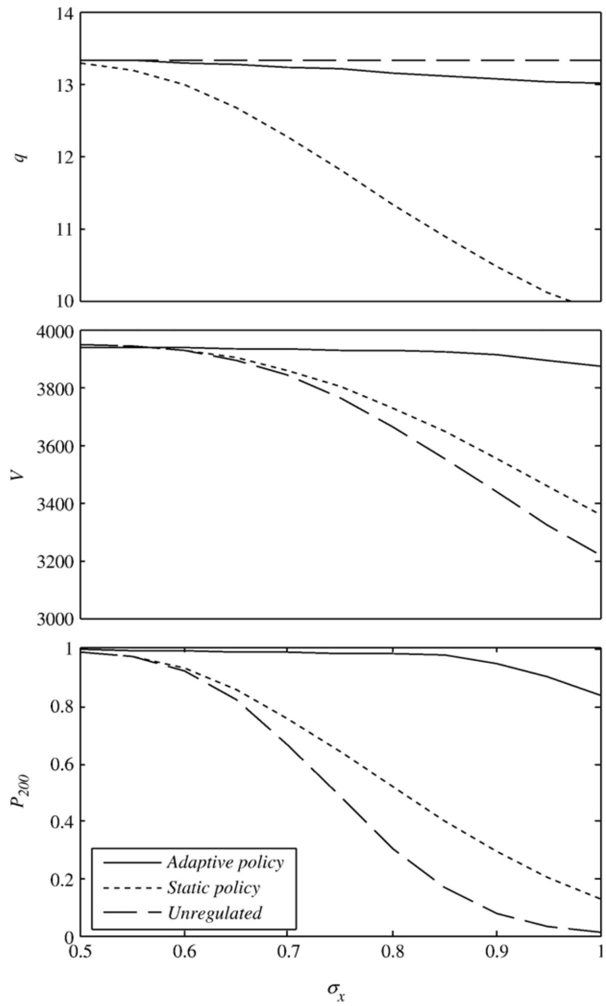

Fig. 2 – Effect of environmental variability on the outcomes from unregulated, static, and adaptive policy approaches, where q is the average loadings cap, V is the expected net present value, and P200 is the probability that the species will survive to the end of the 200 period planning horizon.

To solve for the optimal adaptive policy function, $q_A(y)$, we use backward iteration to obtain convergence of $M_A(y)$ and $N_A(y)$ to stationary functions.[11]

Now we are in a position to illustrate the potential differences between the static and adaptive policy approaches. We first solve the model outlined above with a set of (arbitrarily chosen) baseline parameter values, which give the results shown in Table 2. In this case, with no regulation there is a 49% probability that the stock will survive to the end of the planning horizon (200 periods in this example). The optimal static policy would increase this long-run survival probability to 64% by reducing the loadings level from 13.33 to 11.80 in every period, thereby increasing the expected total net present value from 3763 to 3801. This increase in the total social benefits comes from trading a decrease in market benefits of 30 for an increase in non-market benefits of 68. The optimal adaptive policy reduces the loadings level to (an average of) 13.20, leading to a total net present expected value of 3929 and a long-run survival probability for the endpoint up to nearly 99%. This comes from trading a decrease in market benefits of 10 for an increase in non-market benefits of 176 compared to the unregulated scenario. This is a serious improvement over the static policy.

The reason that the adaptive policy can achieve a higher level of both market output and ecological sustainability than the static policy is because the adaptive approach uses new information on environmental conditions garnered every period, and lowers the loadings cap just enough to protect the stock in those periods when the environmental conditions are especially poor. Conversely, although these very poor periods come relatively infrequently, the static policy must account for the fact that they will come and strikes the best balance possible between the fixed economic output level and the fixed per-period stock survival probability. The resulting optimal fixed loadings cap is highly over-protective (relative to the adaptive policy) in those periods when the environmental conditions are good and slightly under protective in those few periods when conditions are poor.

Of course, by construction the adaptive policy must perform at least as well as the static policy because the latter is a constrained version of the former. The actual difference in performance will depend on the details of each case. To get some feel for the relative performance of the two approaches under different conditions we can perform a sensitivity analysis – varying each parameter in turn while holding the others constant – and plot the results.

This framework distinguishes between environmental variability and measurement error while accounting for both sources of uncertainty simultaneously in the evaluation. Figs. 2 and 3 show that the effectiveness of an adaptive versus a static policy approach depends crucially on the relative magnitudes of these two sources of uncertainty.[12] First, consider the environmental variability σ_x: Fig. 2 shows that increasing σ_x will decrease the value of all three policies, but notice that both the total value and the long-term stock viability using the adaptive policy decrease much less rapidly than when using the static policy. The adaptive approach is relatively robust to uncertainty arising from environmental variability. Fig. 3 shows the effect of measurement error. As σ_y increases, the value of the adaptive policy approaches that of the static policy. This stands to reason because as σ_y increases the measurement instrument becomes less informative and the unconditional prior distribution $f(x)$ is modified less each period as a result. In the limit of a completely uninformative measurement instrument, the static approach is the best that can be done. We also examined how the results are affected by the discount rate assumption and the existence value estimate and found the adaptive policy to be much less sensitive

[11] Since these will converge to stationary functions we have dropped the t subscripts from the y's in Eqs. (3) and (4).

[12] Note that we are using the term "uncertainty" throughout in place of what some authors would call "risk;" we can specify (at least subjective) probability distributions for all variables and parameters in the model.

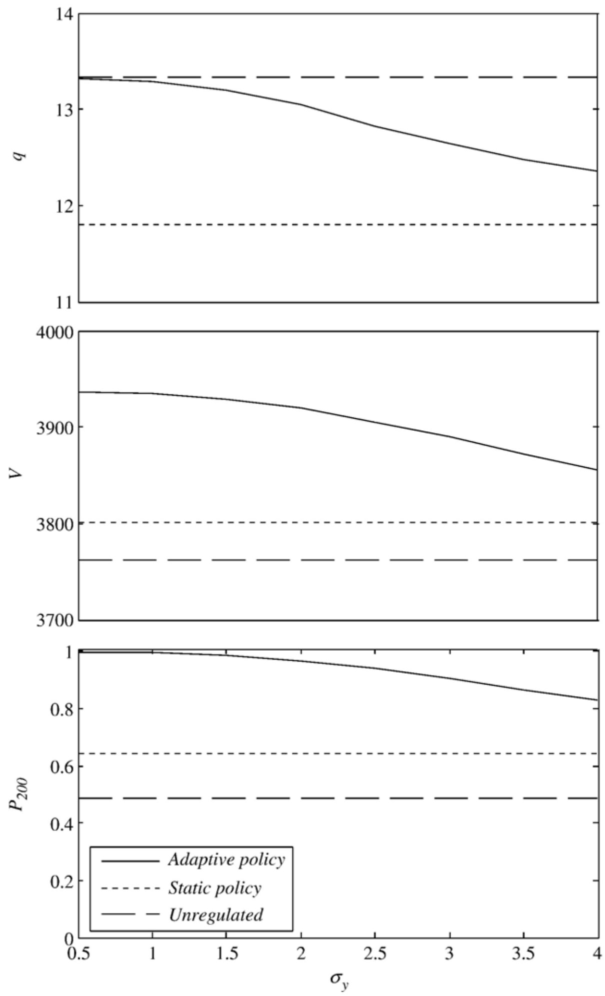

Fig. 3 – Effect of measurement error on the outcomes from unregulated, static, and adaptive policy approaches, where q is the average loadings cap, V is the expected net present value, and P200 is the probability that the species will survive to the end of the 200 period planning horizon.

than the static policy on both counts.[13] In other words, even when the discount rate is relatively high and the existence value is relatively low, the adaptive policy produces a significant increase in the stock survival probability compared to the static policy.

Although our main goal here was to demonstrate how an adaptive management approach can alleviate "policy lock-in," as criticized by Bromley (2007-this volume), our model also bears upon other points in this special section where an adaptive approach to environmental policy assessment is recommended as a potential solution to some of the shortcomings of standard practice. Norton's (2007-this volume) discussion of adaptive management focuses more on learning about and accommodating evolving human preferences; Gowdy (2007-this volume) alludes to this issue as well. Incorporating this aspect of adaptive management into a quantitative framework such as the one illustrated in this section would require developing an explicit model of how υ might change over time, analogous to our model of stochastic pollution impacts. Howarth (2007-this volume) notes that operationalizing the non-declining utility criterion for sustainability requires accurate long-run forecasts of social, economic, and environmental trends, which is difficult to say the least. The way that the adaptive approach deals with these forecasting uncertainties is to maintain a flexible stance and to allow for a wider range of possible ecological outcomes in more distant future periods. The adaptive management approach does not use a long run forecast treated as certain. Instead it uses a stochastic model of ecological dynamics, which by its very nature does not provide "a forecast" in the usual sense. The propagation of inevitable forecasting uncertainties is built into the analysis directly.

6. Conclusions

As we have seen, analysts still often struggle to address ecological sustainability in a rigorous way in environmental policy assessments. Our brief review of seven recent EPA benefit–cost analyses suggests that to date only a limited accounting of ecological benefits occurs in environmental policy assessments. We have discussed two ways that policy assessments might be improved in this area: On one hand, a wide-angle view of the underlying environmental problems could prevent policy inertia that could lead to an irreversible population or ecosystem collapse even when restoration is technically feasible and welfare enhancing. On the other, a more flexible policy stance afforded by keeping a close watch on relevant ecological indicators in principle can increase both ecological sustainability and economic efficiency compared to a traditional static policy approach. Nevertheless, moving forward in either direction will pose non-trivial challenges.

The usual practice of carving up environmental protection into a set of decisions treated independently may be generating sub-optimal results in the aggregate, but it has developed in part because coordination can be onerous. Taking the rationale for coordination to the extreme, a policy assessment for a single source type could require consideration of all directly affected endpoints, the other sources affecting those endpoints, the other endpoints affected by those sources, and so on until considerations extends to the all of the nation's biomes and economic sectors. Just as economists have developed conventions for determining how many indirect market effects to consider in benefit–cost analyses, the insight of ecologists will be required to establish similar conventions for determining just how wide a view should be taken in any particular policy assessment where ecological sustainability is a concern.

Another possible strategy may be to flip the assessment process on its head, by first setting priorities for environmental protection at a large scale and coarse resolution, identifying the general categories of ecological endpoints the public most wants protected, and then determining the most cost-effective

[13] Over the range of ρ between 0.01 and 0.1, the loadings cap and stock survival probability for the static policy range from 10.5 to 12.7 and 0.77 to 0.55, respectively; for the adaptive policy they range from 13.1 to 13.2 and 0.99 to 0.97. Over the range of υ between 25 and 75, the loadings cap and stock survival probability for the static policy range from 12.5 to 11.3 and 0.57 to 0.70, respectively; for the adaptive policy they range from 13.23 to 13.18 and 0.983 to 0.991.

projects to increase protection of those resources given the prevailing budget constraints.[14] The first priority-setting step of this strategy could involve an iterative and deliberative discourse with the public on what aspects of the environment they want protected, i.e., a "futures" exercise. Broad questions about overall priorities may be easier to answer and possibly more reliable than pressing people to consider detailed tradeoffs between unfamiliar ecological endpoints.[15] Second, ecologists and economists would address the technical challenges of quantifying ecosystem resilience and forecasting ecological change and determining the most cost-effective means of achieving the priorities set in the first step. After the cost-effective projects identified in step 2 are underway and measurable outcomes can be observed, the high-level priority setting process of step 1 can be repeated and overall budgets can be reconsidered in light of these outcomes. The frequency at which these steps should be repeated will depend on the rate of change of ecological conditions, technology, human preferences, and our understanding of the dynamics of all of these processes.

On one hand, this approach would require considerably more coordination across agencies and disciplines and currently exists and may require new legal arrangements. On the other hand, the approach could help resolve some of the limitations of institutional compartmentalization and may even cost less in the long run, given the inefficiencies of looking at the same endpoints time and again from slightly different vantages.

While adaptive management has been applied to a variety of ecological control problems, such as fishing and waterfowl hunting (Williams, 1996; Lubow, 1996; Johnson and Williams, 1999; Woodward et al., 2005; Drechsler and Watzold, 2007), the feasibility of this approach at scales relevant to federal regulation remains an open question. A number of EPA regulations contain provisions for updating using the best scientific information currently available at regular intervals. (For example, the Safe Drinking Water Act requires EPA to review each National Primary Drinking Water Regulation at least once every six years, and the Clean Air Act requires review of the standards for particulate pollution every five years.) The general framework illustrated above suggests the need to focus environmental monitoring efforts on valued ecological endpoints and leading indicators of adverse ecosystem changes and to increase the frequency at which regulations are updated.[16]

An important challenge for researchers and analysts will be to identify regulatory situations where an adaptive management approach could profitably be employed (e.g., Cannon, 2005) and then to design an optimal joint monitoring and implementation framework for each case. Another challenge will be to make a concerted institutional shift toward reliance on more flexible economic instruments, such as tradable permits in ecosystem service markets, that should be more conducive to adaptive management than traditional command-and-control regulations, such as technology-based standards. A third challenge will be to minimize the differential cost of an adaptive approach, which is associated , not only requires an on-going monitoring program that is coordinated with the decision-making process, but also may induce additional regulatory uncertainty on firms. In any case, an adaptive management approach should be viewed as one among possibly many policies and they all should be evaluated using a common framework (Failing et al., 2004).

[14] While what we have in mind would also involve stakeholders and the general public, EPA's Unfinished Business report (1987) is an illuminating example of a similar high level prioritization exercise conducted mainly by technical experts.

[15] See Opaluch and Segerson's (1989) work using ambivalence theory to explain hypothetical bias in valuation exercises.

[16] Note that, in the context of clean water, EPA has recently invigorated its monitoring efforts, initiating the "first-ever statistically-valid survey of the biological condition" of the nation's waters (EPA, 2006b). However, it remains to be seen how frequently it will be repeated.

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